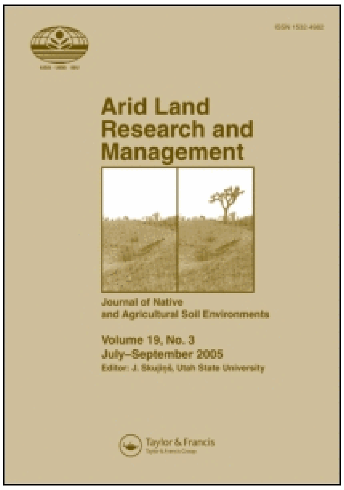

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Rangeland Vegetation and Soil Response to Summer Patch Fires Under Continuous Grazing

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Prescribed fire is used to reduce woody plant and cactus cover and restore degraded rangelands in the southern Great Plains, but little is known regarding the impact of summer fires. We evaluated the effects of summer fires applied as patch burns in continuously grazed rangeland in north Texas. Vegetation and soil responses were measured on patches burned within grazing units in the summers of 1998, 1999, and 2000 relative to that on adjacent unburned control areas in the same grazing units. Annual rainfall during the study was below average for six burns (1998 and 1999) and average or above for three burns (2000). If average rainfall preceded and followed summer burning, degradation was limited to a modest increase in bare ground which recovered to exceed unburned control levels within 2 years. However, when drought conditions preceded and followed burning, there was an increase in bare ground and the proportion of annual forbs and annual grasses at the expense of perennial grasses. These areas took 3–5 years to recover. Areas burned in any year did not recover until after a season of favorable precipitation. The degree of degradation was proportional to the severity of drought conditions. The fires reduced the cover of honey mesquite (Prosopis glandulosa Torr.), other shrubs and cactus (Opuntia spp.), which facilitated herbaceous recovery, mitigating the negative impacts of burning in summer and the increased herbivory on the burned patches. Increases in herbaceous species composition were positively related to woody plant and cactus reduction following fire treatment. Results suggest that summer burning may be an effective and low-cost means of controlling problem plants, increasing pasture heterogeneity, and reducing herbivore impact on intensively grazed patches. However, before the practice of summer burning can be advocated, research needs to determine if post-burn deferment will facilitate more rapid recovery through regulating herbivory after burning to increase the recovery of litter and herbaceous cover and restore desired herbaceous species composition and production.

Keywords brush, cactus, honey mesquite, north Texas, prescribed burning, rangeland restoration

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Many rangeland ecosystems in the southwestern United States have become degraded in the last century in terms of a decline in productivity and biodiversity (Archer and Smeins, 1991; West, 1993; Knopf, 1994), a greater likelihood of irreversible changes in plant species composition (Westoby, Walker, and Noy-Meir, 1989), a reduction in ecosystem resilience (Peterson, Allen, and Holling, 1998), and soil loss (Thurow, 1991). This has occurred largely because of three simultaneous processes: (1) increased woody plant and succulent (cactus) cover; (2) successive replacement of tall- and midgrasses by shortgrasses and then annual grasses; and (3) reduction of litter and perennial grass cover resulting in an increase in bare ground (Ansley et al., 2004; Archer and Smeins, 1991; Collins et al., 1998; Thurow, 1991).

Prescribed fire is the least costly method of reducing woody plant and cactus cover in southern mixed prairie mesquite communities (Teague et al., 2001) and it has been increasingly recognized as an essential part of effective management for grassland and savanna ecosystems (Scifres and Hamilton, 1993; Vermeire, Mitchell, Fuhlendorf, and Gillen, 2004). Although winter fires are safer to use than summer fires, they are less effective in controlling species such as honey mesquite (*Prosopis glandulosa* Torr.) and cactus (*Opuntia spp.*) (Ansley, Kramp, and Jones, 2002). Resource managers in the region have become increasingly interested in summer burns to reduce these plants. In addition, patch burning is a relatively new management technique being implemented where a portion of a pasture is treated to increase pasture heterogeneity (Fuhlendorf and Engle, 2004) and reduce herbivore impact on intensively grazed patches that naturally develop in the landscape from preferential grazing by herbivores (Archibald, Bond, Stock, and Fairbanks, 2005). However, summer burning cannot be widely advocated until the environmental thresholds for safe burns, as well as the ecological and hydrologic impacts of such burns have been adequately quantified (Ansley and Taylor, 2004; Daubenmire, 1968; Daowei and Ripley, 1997).

Wright (1974a) identified soil moisture as the single most critical factor affecting herbaceous recovery after fire in southern mixed prairie communities. It is thought that post-fire herbaceous recovery is slower if fire is applied under dry soil conditions. In addition, warm-season (C_4) grasses are thought to be more susceptible to summer fires because they are physiologically active at the time of burning. Excessive herbivory on the burned areas may also slow recovery of litter and plant cover, and delay or prevent recovery of desired plant species composition and production, and increase soil loss.

From the standpoint of ecological function, the major concerns related to summer burning with no deferment are the removal of protection to the soil from insolation, raindrop action, and overland flow afforded by herbaceous plant cover (Thurow, 1991) during the hottest and driest time of the year (Wright, 1974a), and the loss or decrease of desirable species associated with excessive herbivory on the burned area (Wright, 1974b; Fuhlendorf and Engle, 2004). Degradation associated with excessive herbivory has been well documented (Archer and Smeins, 1991) and involves the progressive replacement of higher seral herbaceous species with lower seral plants, a reduction in herbaceous cover, and an increase in bare ground.

To minimize these negative effects, Wright (1974a) suggested that burned portions of a pasture should be protected by fencing or burning the remainder of the pasture to prevent overuse of the burned portion. Within the mixed-grass prairie region of north-central Texas, known as the "Rolling Plains," the majority of ranchers practice continuous livestock (cattle) grazing and risk resource degradation following summer fire in the absence of grazing deferral. Consequently, the impact of summer burning in conjunction with no livestock deferment requires detailed evaluation before this management practice can be recommended.

In this study, we quantified the impact and rate of recovery of rangeland subjected to summer patch burning under continuous grazing during both dry and wet years.

Methods

The investigation was conducted in the Rolling Plains of north-central Texas (Gould, 1975) on the Waggoner Ranch (33°50′N, 99°5′W) near Vernon. The climate is continental with 220 frost-free, growing days on average. Mean annual precipitation is 648 mm and bimodal in distribution with peaks in May (95 mm) and September (76 mm). Elevation ranges from 335 m to 396 m. Mean monthly temperature varies from −2.3°C in January to 36.4°C in July.

Woody vegetation in the Rolling Plains consists of honey mesquite savanna, lotebush (*Ziziphus obtusifolia* (L.) H. Karst), a shrub of infrequent occurrence, and two species of cactus, tasajillo (*Opuntia leptocaulis* DC.) and prickly pear cactus (*Opuntia phaecantha* Engelm.). The herbaceous vegetation was dominated by a C_3 perennial, Texas wintergrass (*Nassella leucotricha* Trin. & Rupr.), C_4 perennials silver bluestem (*Bothriochloa laguroides* DC.), sideoats grama (*Bouteloua curtipendula* (Michx.) Torr.), meadow dropseed (*Sporobolus compositus* (Poir.) Merr.), buffalograss (*Buchloe dactyloides* (Nutt.) Engelm.), the C_3 annual grasses Japanese brome grass (*Bromus japonicus* Thunb. Ex Murray), and little Barley (*Hordeum pusillum* Nutt.), and the warm season forbs western ragweed (*Ambrosia psilostachya* DC.), annual broomweed (*Gutierrezia texana* (DC.) Torr. & A. Gray) and heath aster (*Aster ericoides* L.). Nomenclature follows Diggs, Lipcomb, and O'Kennon (1999).

Assessment of Degradation and Recovery

To determine the degree of degradation resulting from the summer fires, we measured the amount of bare ground, plant cover, and herbaceous species proportional composition of unburned control areas relative to that of paired treated areas. Degradation (after Archer and Smeins, 1991; Thurow, 1991) was judged to have taken place if one or more of the following changes took place on burned areas relative to the paired, unburned control areas:

1. increase in bare ground
2. increase in forb or annual grass biomass
3. decrease in total grass biomass
4. decrease in perennial grass biomass, comprised of warm season C_4 midgrasses, C_4 shortgrasses, and cool season C_3 midgrasses

Recovery was deemed to have taken place if the reversal of any of the above criteria occurred back to levels of the unburned, paired control areas. Proper grazing and fire management facilitates recruitment and persistence of desired herbaceous species, while poor management results in a decrease of desired species and their replacement by less productive and desirable species and an increase in the amount of bare ground (Archer and Smeins, 1991). In southern mixed prairie communities, the most productive and desirable community is one dominated by C_4 midgrasses with some C_4 shortgrasses and C_3 midgrasses. Poor management and drought conditions lead to progressive deterioration characterized by replacement of taller perennial grasses with shorter perennial grasses, then annual grasses and finally bare ground (Thurow, 1991).

In addition, in order to determine if any long-lasting damage had occurred as a result of the summer burning, we determined whether there were residual differences between paired unburned and burned plots for key soil parameters. These included soil water infiltration rate, aggregate stability, bulk density, and penetration resistance. Resource degradation was adjudged to have taken place if any of these parameters were significantly impaired relative to that of the paired unburned control plots.

Sampling

Sampling sites were located on moderately deep clay loams of the Tillman series (fine, mixed, thermic Typic Paleustoll) derived from Permian red beds with 1 to 3% slopes. In August 2000, August 2001, and October 2003, we monitored vegetation on nine areas that were burned in either the summer of 1998, 1999, or 2000 by lightning-strike wildfires or as prescription summer fire applied by the Waggoner Ranch personnel. The paired controls measured were directly adjacent unburned areas of the same range site, in the same grazing unit. Three fires occurred in each of the three "treatment" years in nine different pastures. The three fires in 1998 were all wildfires that burned an average of 551 hectares; approximately one-fourth of each pasture. The prescribed fires in 1999 and 2000 burned an average of 4% (62 ha) and 2% (55 ha) of each pasture, respectively. Stocking rates in each pasture were 12 ha per AUY^{-1} and grazing was continuous. With this arrangement, fire sites were considered as patch burns within a large (940 to 2957 ha) pasture, and were exposed to cattle grazing immediately after the fire with no post-fire deferment. Burn area size is confounded with the year of burn and precipitation, but we had both large (1998) and small (1999) burns in dry years.

In each pasture, the burn treatment and unburned control sites were sampled by establishing a randomly placed 400-m transect line and measuring herbaceous standing biomass to determine herbaceous species composition to evaluate degradation and recovery as previously outlined. We did not estimate levels of forage utilization. Sampling was conducted along transect lines at 20-m intervals within a 0.05-m^2 quadrat using the dry weight-rank method of Mannetje and Haydock (1963) as modified by Jones and Hargraves (1979) and implemented as outlined by Dowhower, Teague, Ansley, and Pinchak (2001). Grass and forb standing crop were determined gravimetrically from the quadrat clippings. Bare ground, litter, and herbaceous cover were visually estimated in each quadrat for a total of 100%. To simplify interpretation, we report herbaceous biomass by functional group using the following six plant functional groups: annual forb, perennial forb, annual C_3 grass, perennial C_3 midgrass, perennial C_4 shortgrass, and perennial C_4 midgrass.

Woody plant aerial cover estimates were also obtained at each herbaceous sampling point (quadrat placement). Most woody plant cover sampling methods record mean values over the area being sampled, not accounting for information at the scale of pattern created by individual woody plants, which is of prime importance in this study. To overcome this shortcoming, we used a sampling technique developed by

Dowhower, Teague, Gerrard, and Conover (2007) to assess woody plant influence at a single point or a small herbaceous vegetation sample area. Using this method, woody plant canopy was estimated at each herbaceous sampling point (quadrat placement) on a scale of 0–8 in each of the four quadrants. If the height of woody plants within each quadrant was less than 45° vertically from the sample point, woody cover was considered to be minimal and was assigned a score of 0. Otherwise, if the woody cover partially filled the quadrant with an angle of 45° to 75° vertically from the sample point, a score of 1 was assigned. A score of 2 was assigned if the height of woody plants was >75° vertically from the sample point. Summing of the values for all four quadrants provided a score of 0–8, with each rank approximating 12.5% cover. Thus a score of 8 equaled 100% woody plant cover. Comparison of scoring on 22 areas of mesquite cover measured using this method with the line-intercept method of Canfield (1941) provided regression coefficients (R^2) of 0.96 (Dowhower et al., 2007).

In 2004, 1 year after the last annual vegetation sampling was completed, key soil parameters were measured to determine if there were any residual negative effects on the soil. These measures included soil water infiltration, bulk density, penetration resistance, and aggregate stability. Each parameter was measured at each burned and unburned site in each pasture. Within each site, (burned and unburned) soils were sampled at 5 points within mesquite subcanopies (tree) and 5 points in adjacent open grassland (grass). At each sample point, soil penetration resistance was measured (2 per point as above, $n = 20$ for burn and $n = 20$ for unburned paired sites) as described by Herrick and Jones (2002) using the impact penetrometer. Soil aggregate stability at each of these same subsample areas was evaluated as described by Herrick et al. (2001). Soil water infiltration was measured at each of the above sampling points using concentric-ring infiltrometers. A constant water level was maintained in the outer and inner rings by frequently adding small amounts of water until a constant rate of infiltration was achieved as described by Bouwer (1986). Soil bulk density and soil moisture were also measured at each sampling point using 50 mm diameter × 100 mm long soil cores and gravimetric analysis.

Statistical Analysis

Statistical analyses compared the burned to the unburned data in each of the three post-fire sample years using the MIXED procedure with the paired unburned transect as a covariable (SAS Institute, 1990). The random effect was identified as burn treatment × pasture. Least-squared means of the adjusted burn treatment values were compared to paired unburned values.

Paired *t*-tests were run by sample year comparing burn with unburn control transects for all 3 burn year transects combined and by individual burn year transects. Significance was at $P \leq 0.05$ unless otherwise noted.

Results

Precipitation and Temperature

The seasonality of precipitation varied considerably among years even when annual precipitation was similar (Figure 1). Total annual rainfall for 1998 through 2003 was characteristic of drought years with the exception of 2002, which was abnormally

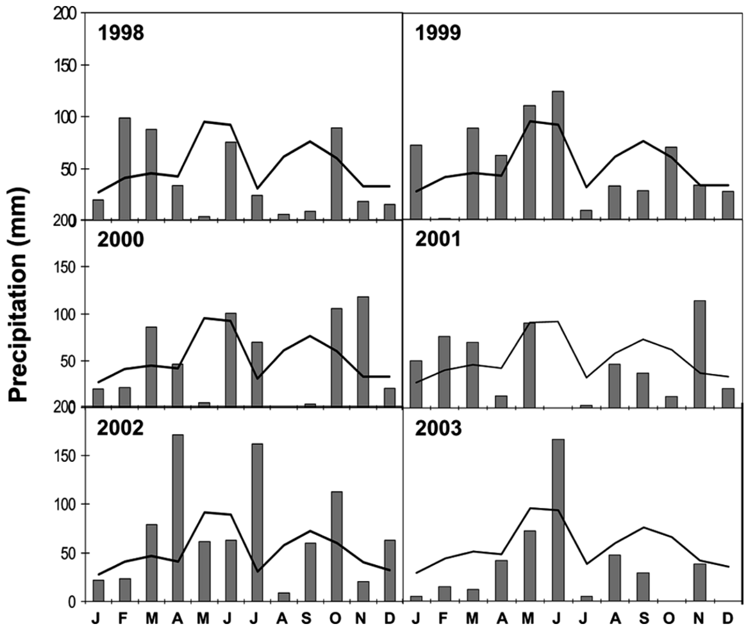

Figure 1. Monthly precipitation (▬) relative to long-term mean monthly precipitation (—) (n = 28 years) at the Lake Kemp Weather Station on the Waggoner Ranch from 1998 to 2003.

wet. In 2000, precipitation was very low during the summer growing season but adequate rains for recovery were received in the fall of that year. In 2001, spring and early summer season rainfall exceeded the long-term mean. The 2002 rainfall also exceeded the long-term mean. Precipitation in 2003 was similar to the long-term means for the summer growing season.

Temperature varied greatly among growing seasons during the study and it is possible that extended high temperatures during the growing season exacerbated moisture deficits. Table 1 lists the annual number of days with elevated temperature (>38°C). Abnormally high indices of elevated temperature days occurred in 1998 and

Table 1. Annual number of days by month that maximum daily temperature exceeded 38°C compared to the 29-year mean of 33

	May	June	July	Aug	Sep	Oct	Year Total	Index*
1998	5	21	28	14	10	0	78	2.36
1999	0	0	10	26	2	0	38	1.15
2000	10	2	17	29	11	2	71	2.15
2001	0	7	26	17	1	0	51	1.54
2002	0	0	4	6	3	0	13	0.39
2003	0	0	18	18	0	0	36	1.09
2004	1	1	10	2	1	0	15	0.45
Mean							33	

*Annual total/33, which is the 29-year mean.

2000 compared to the 29-year average temperature. In 1999, elevated temperature days were slightly above average with an index of 1.15. The number of elevated temperature days in 2002 was below average (index = 0.39) and the rainfall was above normal. In 2003, the elevated temperature index was 1.09 indicating an average year.

Mesquite, Shrub, and Cactus Responses

Summer fires in 1998, 1999, and 2000 resulted in a significant reduction ($P < 0.05$) of mesquite, shrubs, and cactus cover (Figure 2). The 1998 fires reduced these plants the most, but average reduction of mesquite, cactus, and shrub cover by all fires was 40, 71, and 60% for the 1998 burn, 1999 burn, and 2000 burn treatments, respectively. Fire kills only a small percentage of mesquite trees and reductions in tree canopy cover are largely confined to mortality of above-ground organs (Ansley and Taylor, 2004). In southern mixed prairie communities, mesquite cover levels recover to pretreatment levels after fire in 5 to 7 years because of coppicing from below-ground organs (Teague et al., 2001; Ansley, Pinchak, and Teague, 2005).

Indicator Changes in 2000

Data were not collected the first year after the 1998 burn, but the percentage of bare ground was significantly greater in this burn compared to the unburned plots in the second year post-fire ($P = 0.015$ for 2000; Table 2), so we assume

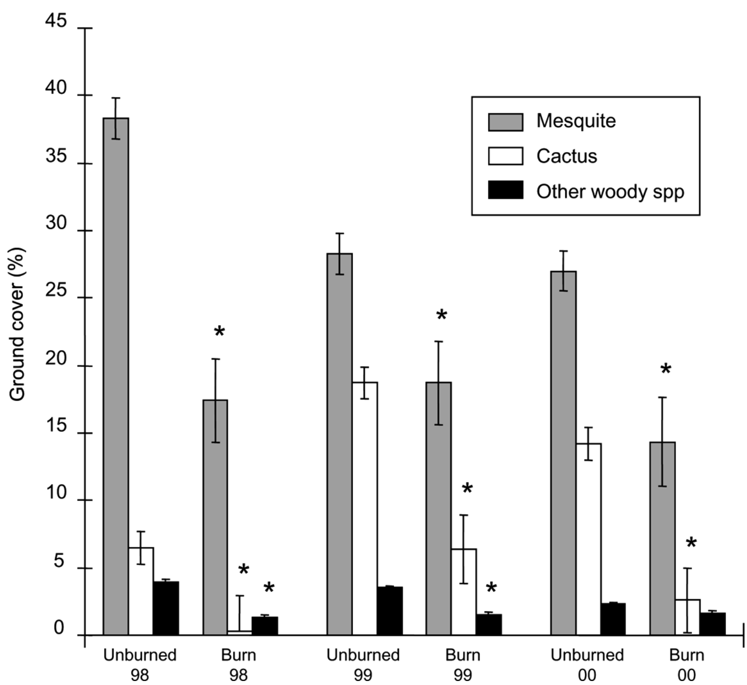

Figure 2. Means of percent mesquite, cactus, and shrub groundcover (%) (±SE) comparing the burned to the unburned plots by year of burn measured in the year of burn in each case. Means differ significantly from unburned control plots for each burn year at $* = P \leq 0.05$.

Table 2. Differences (burned-unburned plots) in bare ground (%) and herbaceous biomass (kg ha^{-1}) in a mesquite grassland in Texas in 2000

Parameter	Burn Year 1998		Burn Year 1999	
	Difference	$P > t$	Difference	$P > t$
Bare ground (%)	+11.4	0.02	+19.2	0.03
Total herbaceous (kg ha^{-1})	−201	0.33	−311	0.11
Grass (kg ha^{-1})	−234	0.30	−306	0.08
Forb (kg ha^{-1})	+33	0.23	−6	0.86
Warm midgrass (kg ha^{-1})	−229	0.04	−189	0.11
Warm shortgrass (kg ha^{-1})	−12	0.52	−139	0.01
Cool midgrass (kg ha^{-1})	−23	0.88	+1.15	0.30
Annual grass (kg ha^{-1})	+31	0.16	−31	0.21

it was also significantly higher than the unburned plots the first year post-fire (1999). Standing crop biomass of warm season midgrasses were also lower in the 1998 burned than unburned plots ($P = 0.04$). Clearly, this burn, which was conducted in and followed by dry conditions caused degradation that was still evident 2 years post-burn.

Similarly, the 1999 burn, which also occurred during and was followed by dry conditions, had increased bare ground ($P = 0.03$) and decreased total grass ($P = 0.08$) and warm season shortgrass biomass ($P = 0.01$; Table 2), indicating significant degradation.

Indicator Changes in 2001

Three years after treatment, the 1998 burn showed some signs of recovery. Bare ground decreased since 2000 and was no different to their respective unburned plots ($P = 0.21$; Table 3). However, forb levels were much higher than in unburned plots ($P = 0.004$) and warm season shortgrasses were lower than in unburned plots ($P = 0.075$), indicating incomplete recovery.

Cover and biomass values for the 1999 burn indicated significant degradation 2 years after the fire. Percent bare ground had improved since the previous year but was still higher than the unburned plots ($P = 0.015$; Table 3). In addition, forb levels were considerably higher for the 1999 burn than unburned plots ($P = 0.012$) and grass biomass was still lower than unburned plots ($P = 0.014$).

One year after treatment, the 2000 burn had more bare ground than unburned plots ($P = 0.029$), while grass and total herbaceous biomass was greater than unburned plots ($P = 0.006$ and $P = 0.057$, respectively). However, most of this was annual grass and cool season midgrass. Although annual grass biomass was the same as unburned plots, cool season midgrass biomass was greater than unburned plots ($P = 0.004$). Shortgrasses did not differ from unburned plots and warm season midgrass was less than unburned plots ($P = 0.062$). Thus, only the increase in bare ground relative to unburned plots can be considered degradation

Table 3. Differences (burned-unburned plots) in bare ground (%) and herbaceous biomass (kg ha^{-1}) comparing the burned to the unburned control plots in a mesquite grassland in Texas in 2001

Parameter	Burn year 1998		1999		2000	
	Difference	$P > t$	Difference	$P > t$	Difference	$P > t$
Bare ground (%)	+3.5	0.21	+5.2	0.02	+3.3	0.03
Total herbaceous	+523	0.10	+712	0.08	+396	0.06
Grass (kg ha^{-1})	+99	0.73	−479	0.01	+408	0.01
Forb (kg ha^{-1})	+424	0.00	+1191	0.01	−12	0.92
Warm midgrass (kg ha^{-1})	−42	0.51	−156	0.14	−106	0.06
Warm shortgrass (kg ha^{-1})	−54	0.07	+8	0.81	−5	0.71
Cool midgrass (kg ha^{-1})	+140	0.13	+149	0.07	+464	0.00
Annual grass (kg ha^{-1})	+55	0.87	−479	0.00	+51	0.56

while the increase in total perennial grass biomass indicates recovery relative to unburned plots only a year after treatment.

Indicator Changes in 2003

By 2003, all burn treatments had recovered in all categories measured. Although the 1998 burn produced less cool season midgrass biomass than unburned plots ($P = 0.09$; Table 4) this did not result in less total perennial biomass ($P = 0.51$). The 1999 burn total grass was higher than unburned plots ($P = 0.09$), indicating recovery beyond the level of the unburned control. For the 2000 burn, all parameters did not differ from unburned plots ($P \geq 0.19$).

Table 4. Differences (burned-unburned plots) in bare ground (%) and herbaceous biomass (kg ha^{-1}) comparing the burned to the unburned control plots in a mesquite grassland in Texas in 2003

Parameter	Burn Year 1998		1999		2000	
	Difference	$P > t$	Difference	$P > t$	Difference	$P > t$
Bare ground (%)	+1.9	0.54	−5.1	0.10	−0.8	0.76
Total herbaceous (kg ha^{-1})	−276	0.51	+225	0.09	+346	0.42
Grass (kg ha^{-1})	−326	0.47	+214	0.09	+496	0.24
Forb (kg ha^{-1})	+50	0.82	+12	0.83	−150	0.29
Warm midgrass (kg ha^{-1})	+151	0.60	−179	0.45	+130	0.54
Warm shortgrass (kg ha^{-1})	−48	0.28	−34	0.77	−40	0.64
Cool midgrass (kg ha^{-1})	−480	0.09	+419	0.12	+402	0.19
Annual grass (kg ha^{-1})	+50	0.30	+9	0.60	−3	0.86

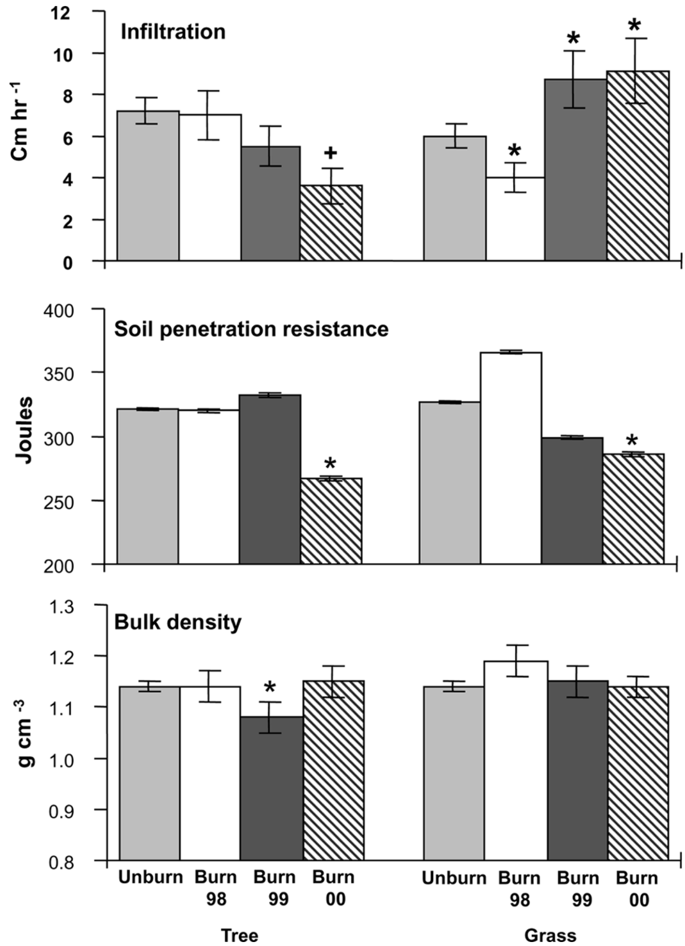

Figure 3. Means of soil water infiltration, soil penetration resistance, and soil bulk density (±SE) comparing the burn plots in subcanopy (tree) or open grassland (grass) to the unburned plots. Means differ significantly from unburned control plots at $* = P \leq 0.05$ and $+ = P \leq 0.10$.

Soil Parameters

There was a significant increase in the soil water infiltration rate in the interstitial grass sites between mesquite canopies with the 1999 burn and the 2000 burn compared to that with the unburned plots in 2004 (5 and 4 years post-fire, respectively) (Figure 3) ($P < 0.05$). In contrast, the infiltration rate under mesquite canopies was significantly less ($P = 0.09$) with the 2000 burn than the unburned plots. Similarly, the soil infiltration rate was also significantly less in the 1998 burn open grassland than the unburned plots ($P = 0.03$). However, infiltration means for the whole area in each case were not less than the unburned plots ($P > 0.10$).

Soil penetrometer resistance was lower in the 2000 burn tree canopy and open grassland sites than the unburned plots. This would normally be associated with increased soil water infiltration but this was only evident with the 2000 burn open grassland sites and the opposite result was measured at the 2000 burn tree canopy sites.

Soil bulk density was lower in the 1998 burn tree canopy and it was less than the unburned plots. It was our expectation that lower bulk density would have been associated with enhanced infiltration but that was not the case with the 1999 burn tree canopy sites. There were no differences in aggregate stability among any of the unburned plots or burn treatments ($P > 0.10$).

Discussion

If average or greater than average rainfall preceded and followed summer burning, degradation was limited to a modest increase in bare ground which recovered to exceed unburned control percentages within 2 years. However, degradation in the form of increased bare ground and forb biomass and a decrease in perennial grasses occurred when summer drought conditions preceded and followed burning. The amount of degradation recorded was proportional to the severity of the rainfall deficit. However, rainfall deficit was not the only factor associated with causing prolonged degradation; elevated temperatures exacerbated the effects of soil moisture deficiency. In addition, the reduction of woody plants and cactus probably facilitated herbaceous recovery, and mitigated the negative impacts of both summer fire and increased herbivory on the burned patches. These results concur with those of Wright (1974a) who found that detrimental effects of winter fires were most likely to occur when drought conditions preceded burning.

Decreases in soil moisture associated with drought have been documented to increase bare ground and decrease herbaceous production and basal area (Anderson, Smith, and Owensby, 1970; Fuhlendorf and Smeins, 1997; Teague, Dowhower, and Waggoner, 2004). Annual forb and annual grass production was proportional to bare ground levels of the previous season. By increasing the amount of bare ground, which facilitates the increase of these annuals, the low rainfall in the drought years of this study indirectly increased both annual forb and grass production at the expense of perennial grass production. Perennial forb production was not affected by fire in this study.

These results also concur with Valone, Nordell, and Ernest (2002) who reported no interaction between grazing and burning on the plant community in an arid grassland under average rainfall conditions. In the present study, even the short-term degradation caused by summer fires under drought conditions was not detected 5 years post-fire compared to an unburned, grazed control. There were no significant reductions in the perennial C_4 midgrass standing crop 3 years after burning even under summer drought and elevated summer temperatures. Although perennial C_4 midgrass production in burned treatments was reduced relative to the unburned plots the year after burning in drier conditions, they recovered within 1 to 3 years.

Recently-burned areas usually receive higher levels of herbivory than the surrounding vegetation (Bailey etal., 1996; Fuhlendorf and Engle, 2004; Archibald et al., 2005), and periods of below-average rainfall compound the impact of such selection on vegetation and soils (Teague et al., 2004). If threshold amounts of biomass and litter are not maintained, a degradation spiral is initiated (Thurow, 1991). In this study, we could not identify the effects of post-fire selective herbivory without an ungrazed control or a treatment that included grazing deferment. Although we did not measure utilization differences between burned and unburned areas, we did observe a greater concentration of grazing animals on burned patches beyond the year of burning. We believe that increased herbivory on burned patches may have contributed to resource degradation. This herbivory effect needs to be researched in this environment to determine expected ranges of degradation that may occur when practicing summer burning and if post-fire deferment would facilitate recovery.

Fuhlendorf and Engle (2004), when burning one-sixth (17%) of the total grazing unit, found that cattle in a successively burned pasture devote 75% of grazing time to one-third of the area burned the previous year. Clearly, the proportion of burned patch in a grazing unit merits further research in order to determine resource recovery and sustainable thresholds. Our study emphasizes the need to include how climatic conditions influence the recovery process. The size of burned patches in our study differed mostly between the 1998 fires, where patches composed 26% of the total grazing unit, and fires from the other years studied in which patches made up 3% of the total grazing unit. The opportunistic nature of our study did not allow us to evaluate the effects of different burned-patch sizes in relation to total grazing unit size.

Management Implications

These results present a worst case scenario for summer burning in that only a small portion of each pasture was burned with no post-burn deferment as advised by Wright (1974a) or a reduction in herbivory on burnt patches by successively burning other areas in the same grazing unit as advocated by Fuhlendorf and Engle (2004). Although the areas burned in 1998 and 1999 recovered from negative fire effects after 3 to 5 years, these results indicate that the practice of summer burning with no grazing deferment does slow recovery of the resource if conducted during, and followed by, drought conditions relative to doing so under favorable precipitation conditions. Clearly, areas burned in any year did not recover until after a season of favorable precipitation. During periods of drought this may take many years. Since permanent negative effects were not measured when climatic conditions were conducive to rapid post-fire recovery, the practice of burning in summer should not be condemned since it is an effective and low-cost means of controlling problem plants (Wright and Bailey, 1982; Ansley et al., 2002; Ansley and Taylor, 2004), increasing within pasture heterogeneity (Fuhlendorf and Engle, 2004), and reducing the herbivore impact on intensively grazed patches (Archibald et al., 2005). However, before the practice of summer burning can be advocated, research needs to determine if post-burn deferment will facilitate more rapid recovery through regulating herbivory after burning to increase the recovery of litter and herbaceous cover and restore desired herbaceous species composition and production.

Although this study indicates that there was recovery within 5 years of all criteria measured after summer patch fires, we should be wary of the results of such short-term studies. To effectively control woody plants and cacti, fire must be applied regularly (Hamilton and Ueckert, 2004) at 6 to 7-year intervals in the southern mixed prairie communities (Ansley and Taylor, 2004). There are no long-term experiments in this eco-region to indicate whether burning at such frequencies would maintain or improve the health of this ecosystem in the long-term. If fire is used in these communities to regularly reduce mesquite and cacti, managers need to pay careful attention to stocking levels (Higgins, Kantelhardt, Scheiter, and Boerner 2007; Teague et al., 2008) and monitor recovery of key parameters to ensure there is full recovery after each burn before applying further burns to the same area.

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